

# FINAL REPORT

Long-term effects of restoration treatments in a  
Wyoming big sagebrush community invaded by  
annual exotic grasses

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16-1-03-25

**May 2020**

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## **Key Words**

Invasive species, exotic annual grass, sagebrush community, functional groups, juniper removal, seeding, restoration

## **Acknowledgements**

Special thanks to Ariel Muldoon (Oregon State University), the Krawchuk lab (Oregon State University), and Dr. Marti Anderson (Massey University). Special thanks to Mark Lesko, Ann Roberts, and Steve Gibson of the Ochoco National Forest (ONF) and Crooked River National Grassland for providing input and support of the initial study. Thanks to the slash pile burn crew led by ONF's Brian Shultz. Manuela Huso (Oregon State University) and Patrick Cunningham (USDA Forest Service, Pacific Northwest Research Station) provided experimental design and statistical consulting. Richard Halse (Oregon State University Herbarium) provided expertise in sagebrush identification. Thanks to Troy Maddux, Steve Swenson, Julie Nist and Kent Coe for field support.

## **Data Availability Statement**

Data are archived at the US Forest Service Research Data Archive, Kerns and Day 2020

<https://doi.org/10.2737/RDS-2020-0032>

## **Abstract**

Western US sagebrush ecosystems are threatened due to multiple interacting factors: encroachment by conifer woodlands, exotic annual grass invasion, severe wildfire, climate change, and anthropogenic development. Restoration of these communities is primarily focused on reducing conifer species such as western juniper, with the goal of increasing native herbaceous perennials and sagebrush and decreasing exotic annual grass invasion. Assessing the long-term success of restoration treatments is critical for informing future management and treatment strategies since short-term patterns do not generally predict long-term trends. Using a designed experiment from a Wyoming big sagebrush community that was established in 2008, we examined the long-term vegetation response to juniper removal and seeding (cultivar and local) in disturbed and undisturbed areas (slash pile, skid trails, no disturbance). We also examined the landscape scale plant response to juniper removal using repeatedly measured randomly located transects across two restoration units. We found that seeded species persisted in the long term and also mitigated exotic grass increases. Seed mix source often did not matter. Examination of the landscape scale response to juniper cutting revealed that restoration treatment goals were not met. More aggressive pre- and post- treatment exotic grass control methods may be needed to meet restoration objectives in Wyoming big sagebrush landscapes invaded with annual grasses.

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## Objectives

Sagebrush ecosystems stretch across a large portion of the western US and are threatened by the encroachment of woody species, invasion of exotic annual grasses, increasing frequency and severity of wildfire, anthropogenic development and climate change (Miller et al., 2011). In 2008, we initiated a study to examine the response of vegetation to a western juniper removal project in an exotic grass invaded Wyoming big sagebrush community in central Oregon. Areas highly disturbed by treatment operations were seeded with native species post-treatment (JFSP Project ID: 05-2-1-05; Kerns and Day, 2014). Prior to treatment, the project area had infestations of some of the most invasive annual grasses in western North America: medusahead (*Taeniatherum caput-medusae*), cheatgrass (*Bromus tectorum*) and North Africa grass (*Ventenata dubia*). Short-term results were reported in Kerns and Day (2014).

The objectives this study were to assess the longer-term (~8 years) effectiveness of both seeding and juniper removal more generally in terms of restoring sagebrush community composition and structure and increasing ecosystem resilience. First, we remeasured and analyzed data from slash piles and skids trails that were experimentally seeded in the original study (Kerns and Day 2014). Specifically, we wanted to determine if: a) seeded species persisted and facilitated broader community recovery, b) there was evidence of delayed exotic species suppression in seeding treatments, c) cultivar and local seed sources performed similarly (e.g. total cover, effects on invasive species), d) disturbance conditions impacted seeding effectiveness in the long term and e) short-term results were predictive of long-term restoration outcomes. Secondly, we wanted to determine if key goals for sagebrush community recovery and maintenance were met more generally across the landscape by remeasuring permanent transects across the study site established prior to juniper removal. Key restoration metrics we were able to examine included: a) total cover and cover of sagebrush and perennial bunchgrasses and forbs, including bluebunch wheatgrass, b) cover of the shallow rooted perennial Sandberg bluegrass (*Poa secunda*), and c) plant richness. For both objectives, we also hypothesized that the variability in pre-cut juniper tree abundance would influence the long-term vegetation response owing to pretreatment understory species composition and the degree of disturbance due to cutting (Kerns and Day 2014). Throughout this report, we refer to these studies as 1) long-term post-treatment seeding effectiveness, and 2) long-term restoration effectiveness.

## Background

Western juniper (*Juniperus occidentalis* Hook.) has undergone an unprecedented expansion in terms of both density and range, specifically in the Intermountain Region (Miller and Rose 1995). The expansion has been extensively studied in the past century (Burkhardt and Tisdale 1969, Eddleman 1987, Miller and Rose 1995) and has been largely attributed to a number of interacting factors including maturity of stands established in climatically favorable conditions, increased atmospheric CO<sub>2</sub>, fire suppression and livestock grazing (Young and Evans 1981, Miller et al. 1987, Knapp and Soule 1998, Miller and Rose 1999). The expansion of western juniper into previously sagebrush-dominated ecosystems is of particular concern as it can lead to a type conversion from shrubland to woodland (Miller et al. 2005, Weisberg et al., 2007; Romme et al., 2009). As a result, over 350 sagebrush dependent plants and animals have been classified as species of concern (Suring et al. 2005b, Wisdom et al. 2005). Historically, sagebrush communities once occupied over 62 million hectares but have decreased by over 40%, making it one of the fastest declining vegetation types in the western US. (Davies et al. 2011).

In addition to the negative impact of western juniper expansion on sagebrush habitat, expansion of western juniper has also been linked to increasing woody fuels and crown fire potential (Miller et al. 2005), and altering ecosystem functioning through changes in hydrological processes (Kormos et al. 2016). Owing to the potential for these negative effects, land managers have sought to remove or reduce western juniper through mechanical cutting and burning in an effort to restore native habitat and decrease fuel loads while working to mitigate the likelihood of invasion of exotic species post removal.

Over a decade ago, the US Forest Service, in collaboration with other private and federal landowners, implemented a restoration project on the Crooked River National Grassland (CRNG) in central Oregon. The CRNG is one of the largest tracks of preserved grassland in what once was historically a Wyoming big sagebrush-grassland community. Local managers reported that the encroachment of western juniper had altered the vegetation community composition, decreased wildlife habitat, and increased the risk of wildfire to adjacent homes in the Wildland Urban Interface (WUI). The CRNG project was designed to reduce tree density to pre-European settlement levels through mechanical removal of juniper, with the goal of improving wildlife habitat and restoring the vegetation community composition to more historical conditions by increasing sagebrush and herbaceous perennial plant cover. However, disturbances associated with restoration (canopy removal, ground disturbance, burning) can also have negative impacts on vegetation due to plant mortality, soil compaction, and exotic plant invasion. Therefore the extent to which restoration goals are met can be unclear, particularly in woodlands invaded by exotic annual grasses.

Seeding has been used extensively after juniper removal with an aim to facilitate cover of native or desired species and decrease the probability of establishment of non-native species through competition (Robichaud et al. 2000, Beyer 2004, Urza et al. 2019). Seeding has been successful in some cheatgrass dominated regions (Ratzlaff and Anderson 1995, Cox and Anderson 2004). However, both the species selected for seeding and potentially the origin of seeds can influence restoration outcomes. Perennial grasses are one of the only functional groups to effectively displace invasive exotic species (DiTomasco 2000, Davies 2010). Additionally, while locally derived seed stocks are recommended for restoration to increase the likelihood that plants are adapted to the site (Bischoff et al. 2008), due to their limited availability and high cost (Smith et al. 2007), land managers typically use seeds from off-the-shelf cultivated plant varieties (cultivars).

We previously examined the short-term effect of seeding treatments in ground disturbed areas (slash piles and skid trails) associated with the CRNG project areas in an effort to suppress exotic weeds and enhance native plant diversity post treatment (Kerns and Day, 2014, JFSP Project ID: 05-2-1-05). We reported that in 2011, seeding increased the total cover and richness of native species on slash piles but not on skid trails. In addition, seeding did not reduce the cover of exotic grasses in the short term. However, assessing the long-term success of seeding treatments and juniper removal is critical for informing future management and treatment strategies since short-term patterns do not generally predict long-term trends (Bates et al. 2005, Rinella et al., 2012, Uzra et al. 2019). This is particularly true in areas invaded by exotic annual grasses post-removal of dominant overstory species. In a 25-year restoration study, Copeland et al. (2019) found that five years post-treatment results underestimated the longer term suppressive effect of seeding on non-native exotic cover. Similarly, in a long-term remeasurement study, exotic suppression was not detected until 15 years post seeding (Rinella et al. 2012). Therefore,

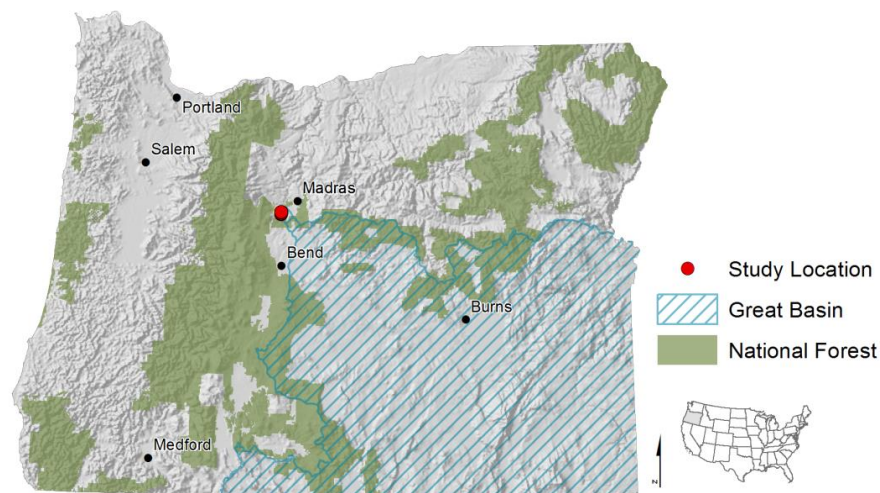
we proposed and were funded to remeasure these seeded areas in 2017 and report on long-term outcomes.

While understanding the long-term trends associated with seeding in disturbed areas is critical, that study was not designed to provide information about the effects of juniper removal and restoration outcomes more generally across the landscape, which is critical for managers to be able to “monitor and adapt for success” (Rangeland Fire Task Force 2015). Therefore we also proposed remeasuring systematic baseline condition transects that were established in 2008 in two separate units in the project area to assess whether key metrics related to sagebrush community recovery and maintenance were met.

## Materials and Methods

### Study Area and Experimental Design

The Crooked River National Grassland Westside Wildland Urban Interface Fuel Reduction Project is a 720-acre allotment within the 155,000-acre Crooked River National Grassland site located along the convergence of the Deschutes, Crooked and Metolius Rivers (CRNG; lat 44831026.2200N, long 121820001.0400E) (Fig. 1). The study area is a big sagebrush woodland (*Juniperus occidentalis*/*Artemisia tridentata*/Thurber’s needlegrass (*Achnatherum thurberianum*)) managed by the US Department of Agriculture (USDA) Forest Service, located at a mean elevation of 823 m. Subspecies identification of big sagebrush was not clear based on morphological traits, although habitat and elevation conditions suggest that *wyomingensis* may be the correct subspecies (Richard Halse, Oregon State University, personal communication, October 2013). The study area was grazed by sheep up until the 1980s, but since then no livestock grazing has occurred. Mean water year annual precipitation (MWYAP; October–September) was 215 mm for a 27-yr period (1985–2012; based on Haystack, Oregon RAWS USA Climate Archive, 18 km southwest of the study area (Western Regional Climate Center 2013). More details on the study site can be found in Kerns and Day (2014).



**Fig. 1.** Location of the study area in central Oregon relative to national forestlands and the northwestern edge of the Great Basin as mapped by the Great Basin Restoration Initiative (USDI BLM, 2015).

Mechanical tree removal was completed in 2008 – 2010 by cutting most post-settlement juniper by chainsaw and removing material by skid-steer (Table 1). About 20% of the post-settlement juniper was retained in patches in each unit and these areas were used as control areas (Fig. 2). Control selection also focused on protecting wildlife habitat and old-growth retention (old-growth juniper trees were not cut and were identified by characteristics such as twisted, gnarled trees) and was done in collaboration with stakeholders, thus controls are not completely random (Fig. 2). Since juniper abundance varied across the study area in a patchy manner, we categorized areas into low (~13%) and high (~ 47%) cover prior to establishing plots and transects.



**Fig. 2.** In the distance a remnant patch of juniper in 2011 (after cutting) is shown. These areas were embedded within the restoration project and were used as controls. They are referred to as “no treat” in graphs and tables.

Table 1. Timeline of activities for this study. Data collection was funded by a prior JFSP project 2008-2011. Data collection in 2013 was funded by the Pacific Northwest Research Station. Data collection in 2017 was funded by this project. All data have been archived and are accessible online (Day and Kerns, 2020).

<b>Activity</b>	<b>Date</b>
Control units established	April 2008
Slash-pile plot locations established	April 2008
Slash-pile pretreatment data collected	Spring/Summer 2008
Transect data collected	Spring/Summer 2008
Juniper thinned	Spring - Fall 2008
Slash piled by fire crew	Fall 2008
Trees removed from site	Fall 2008 – Summer 2009
Skid trail plots established (no pretreat data)	May 2009
Piles burned	December 2009
Piles seeded	December 2009
Skid trails raked and seeded	December 2009
Post treatment data collection	
Slash piles and skid trails	Spring/Summer 2010, 2011, 2013, 2017
Transect data	Spring/Summer 2013, 2017

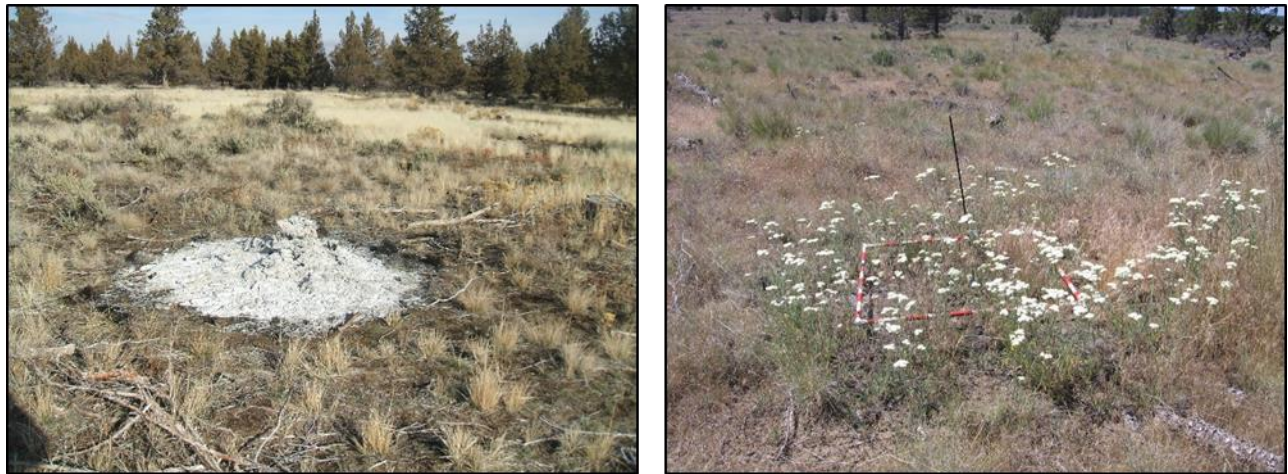


### *Long-term seeding effectiveness*

For both ground-based disturbance experiments, two seed mixes were tested, a cultivar and a local mix. The local mix used seeds collected by managers within the CRNG at elevations and soil types that were similar to the study area. Species used in the mix were: bottlebrush squirreltail (*Elymus elymoides* or ELEL), bluebunch wheatgrass (*Psuedoroegneria spicata* or PSSP), and western yarrow (*Achillea millefolium* or ACMI). Squirreltail is a short-lived early seral native perennial bunchgrass, and common yarrow is a rhizomatous native forb known to spread rapidly in disturbed areas (Alekssoff 1999). Bluebunch wheatgrass is a large deep-rooted native perennial and is a target species for restoration in the area. The cultivar mix was created by using the same species, but seeds were not locally sourced and an available off the shelf cultivar mix was used. The names of the cultivars used were PSSP “Anatone”, ELEL “Toe Jam Creek” and ACMI “Eagle Mountain.” Seeding rates approximated those recommended by Sheley (2008) as described in Kerns and Day (2014).

Seeding treatments were randomly assigned to slash piles and skid trail plots as described below. Twenty plots were established prior to juniper removal that served as control plots or no treatment (no trt) where juniper was not removed and no seeding was done. In summary, there were three treatments for each experiment in addition to the no treat: no seed, local seed mix (local), and cultivar seed mix (cultivar).

Slash pile plots. We established and permanently marked random plot center locations in 2008 on the ground where 2-m-diameter slash piles would be burned after cutting (Fig. 3). We randomly selected an equal number of points within each juniper canopy cover class (low, high) for each treatment in a GIS. Slash was hand-piled into small piles and overwintered before burning in December of 2009. Broadcast seeding onto snow for both slash piles and skid trails (see below) was completed in winter 2009. The slash pile experiment was a balanced design, with 20 replicates (20 no treat and 20 plots each of no seed, cultivar and local).



**Fig. 3.** Slash piles after burning (left) and after seeding and several years of recovery (right). The white flowered plant on the right is common yarrow (ACMI), one of three seeded species, and the reddish grass is cheatgrass, the most common exotic annual grass in the study area. A one meter sampling quadrat is shown on the right.



**Skid trail plots.** Skid trails were formed during the fall and winter of 2008 by skid steer (Fig. 4). Skid trail plots were randomly selected and established for seeding in the spring of 2009. Trails were divided into 20-m segment lengths and each segment was randomly selected in GIS. Placement of plots within the skid trails was constrained to areas with clear evidence of skid trail usage, exposure of bare soil for seeding and > 25 m from another plot. There were 15 replicates for each seeding treatment (no seed, cultivar and local). The experiment is robust but unbalanced (15 plots each no seed, cultivar, local and 20 no treat).



**Fig. 4.** Skid trails shortly after juniper removal (left) and after several years of recovery (right).

Due to the contract time period, some skid trails were formed at least a year before they were seeded. Seeded species were not as successful in these areas as compared to the slash piles (Kerns and Day 2014).

#### *Long-term restoration effectiveness*

Prior to cutting in 2008, 166 30 m-long point transects were systematically established across the study area (Fig. 5). After treatment these transects included areas that were not cut, and less commonly, crossed skid trails and slash piles. To gain more insight into the removal treatment, we separated transects into two distinct groups or treatments for analysis: juniper cut ( $n=142$ ) and no cut ( $n=24$ ). No cut transects were reduced due to the small amount of the landscape that was not treated based on the project contract. Transects which bisected both cut and no cut areas were truncated to include only the no cut portion of each transect (plus a one-meter buffer). Note that no cut transects were nested within the larger restoration project area (Fig. 2).

### **Sampling**

#### *Slash pile and skid trail plots.*

Data were collected using the same methods across all years for both slash piles and skid trails (see Kerns and Day (2014) for additional details). Cover data were collected by species using a square 1 m plot frame located at the plot center. Plant cover is the percentage of ground area beneath the aerial canopy of a given species or life form. Cover was visually estimated to the nearest percentage using systematic marks on a plot frame. Half percent designations were used up to 3% cover. Any species less than 1% was recorded as 0.5%. Throughout the entire

project only one person recorded cover; although in 2017 an additional observer recorded cover and systematic training was completed to standardize cover measurements. Cover was also recorded for bare soil, rock (>2 mm), and litter (all dead plant material, e.g., pine needles, bark, and dead grass). Mature trees (>1.37 m tall) and all stumps were tallied, and diameter at breast height or two perpendicular stump diameters were recorded.



**Fig. 5.** Repeatedly measured transects through time can reveal insights about the juniper removal treatment across the landscape and long-term restoration effectiveness. This photo was taken in 2017 in an area heavily invaded with annual grass.

### *Transects*

Data were collected using 30-m long point transects within each unit using methods based on Herrick et al. (2005) (Fig. 5). Vascular plant presence by species was recorded every meter, along with ground cover type (bare soil, rock, etc.). Because they are small in stature and patchy in nature, noxious weeds were recorded every 1 cm. Noxious weeds included medusahead, North Africa grass, bull thistle (*Cirsium vulgare*), and spotted knapweed (*Centaurea stoebe* subsp. *micranthos*). Bull thistle and spotted knapweed were exceptionally rare. Cheatgrass was not included in the more frequent point measurements as it was not originally a target of interest in the Environmental Impact Statement, and it occurs frequently enough to be well represented on the meter point measurements.

## **Data Analysis**

### *Long-term seeding effectiveness*

Cover of each seeded species in 2017 was analyzed separately as a response variable (PSSP, ELEL, and ACMI). Cover of other species was combined into similar functional groups based on prior work (Kerns and Day 2014) (Table 2). Functional groups were based on life history, morphology, and origin. Sandberg bluegrass was analyzed separately to align with other regional studies, and because of its unique early phenological development, small stature and rooting depth (Davies and Dean 2019). Functional groups used to analyze richness were broader, and included perennial bunchgrasses, perennial forbs, native annual forbs, shrubs and all exotic species.

Response variables were analyzed with a completely randomized analysis of covariance. General linear mixed models were run in R (version 1.1.423, R Development Core Team 2018)

using the nlme (Pinheiro et al. 2019) and emmeans packages (Lenth et al. 2019). Although plots were stratified by pre-treatment juniper cover (high, low), actual juniper abundance (basal area) prior to cutting was used as a covariate. Treatment and juniper basal area, and the subsequent interaction were fixed effects. Statistical significance is discussed where P values are less than 0.10, although we avoid strict dichotomous distinctions and investigate post-hoc comparisons regardless of global ANOVA p-values as apriori treatment contrasts are of interest.

**Table 2.** Understory plant functional groups used as response variables in the seeding effectiveness study, the short code used in graphs, and the dominant species within each group. Origin is native unless noted in the group name. Seeded species were analyzed separately (ELEL, ACMI, PSSP). Nomenclature follows Meyers et al. (2015) for grasses, and Hitchcock and Cronquist (1973) for other species.

Functional Group	Code	Dominant species
Total plant cover	Total	Sandberg bluegrass, Thurber's needlegrass, Idaho fescue
Perennial large bunchgrasses	PBG	No seeded species, Thurber's needlegrass, Idaho fescue*
Shallow, early season grass	POSE	Sandberg bluegrass
Perennial Forbs	NPF	Low pussytoes ( <i>Antennaria dimorpha</i> (Nutt.) Torr. & A. Gray), velvet lupine ( <i>Lupinus leucophyllus</i> Douglas ex Lindl.)
Native annuals	NAF	Pacific popcornflower ( <i>Plagiobothrys tenellus</i> (Nutt. ex Hook.), small fescue ( <i>Vulpia microstachys</i> (Nutt.) Munro)
Exotic grasses	EG	Cheatgrass
Tall annual exotic forbs	TAEF	Tall tumbled mustard ( <i>Sisymbrium altissimum</i> L.), littlepod false flax ( <i>Camelina microcarpa</i> Andr. ex DC.)
Small annual exotic forbs	SAEF	Jagged chickweed ( <i>Holosteum umbellatum</i> L.), spring draba ( <i>Draba verna</i> L.)
Shrubs	SHRUB	Antelope bitterbrush, Wyoming sagebrush, green rabbit brush

\*For the restoration effectiveness analyses, the PBG group includes bottle brush squirreltail (ELEL) and bluebunch wheatgrass (PSSP).

### *Long-term restoration effectiveness*

Functional group categories were similar to those above (Table 2) except the PBG category included PSSP and ELEL (Table 2) and western juniper (JUOC) cover was assessed. POSE was separated as described above. Data from 2008, 2013, and 2017 were used in a repeated measures general linear mixed model to test for the effect of juniper removal (cut, no cut) on functional group cover and richness. To conduct multiple comparisons of linear mixed models, we used the *glht* function in the multcomp package (Hothorn et al., 2008). We investigate post-hoc comparisons of interest regardless of global ANOVA p-values. However, not all contrasts were of interest. The main apriori contrasts presented here are: 1) pretreatment differences (cut – no cut in 2008), 2) short-term treatment comparison (cut – not cut in 2013), 2) long-term treatment comparison (cut – no cut in 2017), and 3) long-term comparisons within treatment (cut 2008 – 2017; no cut 2008 – 2017). Linear mixed models were run in R (version 1.1.423, R Development Core Team 2018) using the nlme (Pinheiro et al. 2019) and emmeans packages (Lenth et al. 2019).

## Results and Discussion

### Long-term seeding effectiveness

#### *Slash piles*

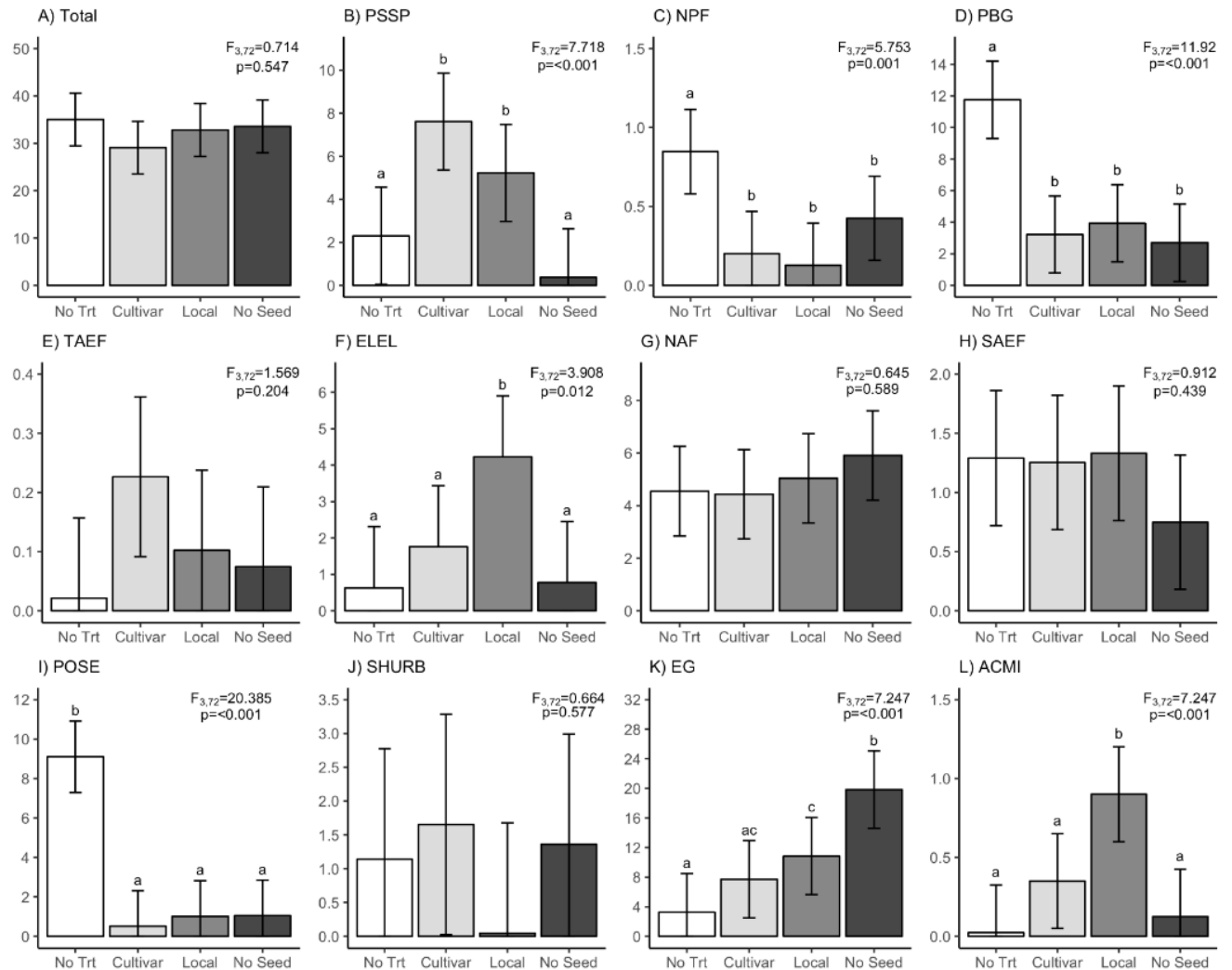
We found strong evidence for a long-term effect of juniper removal followed by slash pile burning and seeding for most plant groups by 2017, including the three seeded species, PSSP, ELEL, and ACMI, as well as for the functional groups NPF, PBG, POSE, and EG (Fig. 6). However, unlike the 2011 short-term results previously reported (Kerns and Day (2014), hereinafter referred to as previously reported or short-term results), there was little evidence for a treatment effect on total plant cover at this point in time. Several native perennial functional groups still had lower cover in areas that were treated, including NPF, PBG, and POSE, although they had increased in cover as compared to results from 2011, although direct statistical comparisons were not made. Conversely, there was little evidence the treatments affected short-lived annual forbs (NAF) or shrub cover, although shrubs had recovered enough to be detected (Fig. 6).

In 2011, pretreatment or precut juniper basal area (PJBA) was significant for Total, POSE, Annual (analogous to NAF), and EG. Total, Annual, and EG cover were negatively associated with higher PJBA, and POSE was positively associated with higher PJBA. However, there was little evidence of an interaction between PJBA and any treatment. In 2017, fewer plant groups showed evidence of a relationship to PJBA. PSSP was positively associated with high PJBA ( $P = 0.02$ ), and EG retained a negative relationship with high PJBA ( $P = 0.08$ ). Again, there was little evidence of an interaction between PJBA and any treatment (Fig. 6).

There is strong evidence that seeding had a long-term impact on the cover of seeded species (PSSP, ELEL and ACMI, Fig. 6). For PSSP, cover was about 3-7% higher in plots seeded with both mixes as compared to the no trt and no seed plots. As noted, PSSP cover was positively correlated with high PJBA ( $P = 0.02$ ), and while this pattern was not found in earlier results, it was evident prior to treatment (Kerns and Day 2014) suggesting some recovery. Cover of both ELEL and ACMI were higher in the local seed mix plots as compared to the no trt and no seed plots, but this trend was not found for the cultivar seed mix. While statistically significant, the differences in cover for ACMI were quite small. Thus, the local seed mix performed better in the long term for the early successional species (ELEL, ACMI), but both seed mixes were equally effective for PSSP. There was little evidence seeding impacted any other native plant functional group.

Exotic grass (EG) cover increased in response to this disturbance combination, with an average of 16% increase in areas that were not seeded (Figs. 6 and 7). Interestingly, unlike the short-term results, both seed mixes lowered EG cover compared to no seeding (ca. 10% cover vs. 20%), but EG cover remained the lowest where there was no disturbance (no trt). TAEF cover was non-existent prior to juniper removal and by 2017 was only observed in trace amounts across the treatments indicating an ephemeral response to disturbance, and SAEF cover was also relatively low. This functional group negatively responded to disturbance and seeding in short-term responses, but by 2017 had recovered to levels in the no trt.

There is some evidence slash pile burning and seeding increased total and PBG richness slightly (Fig. 8) in the long-term as compared to no seeding.



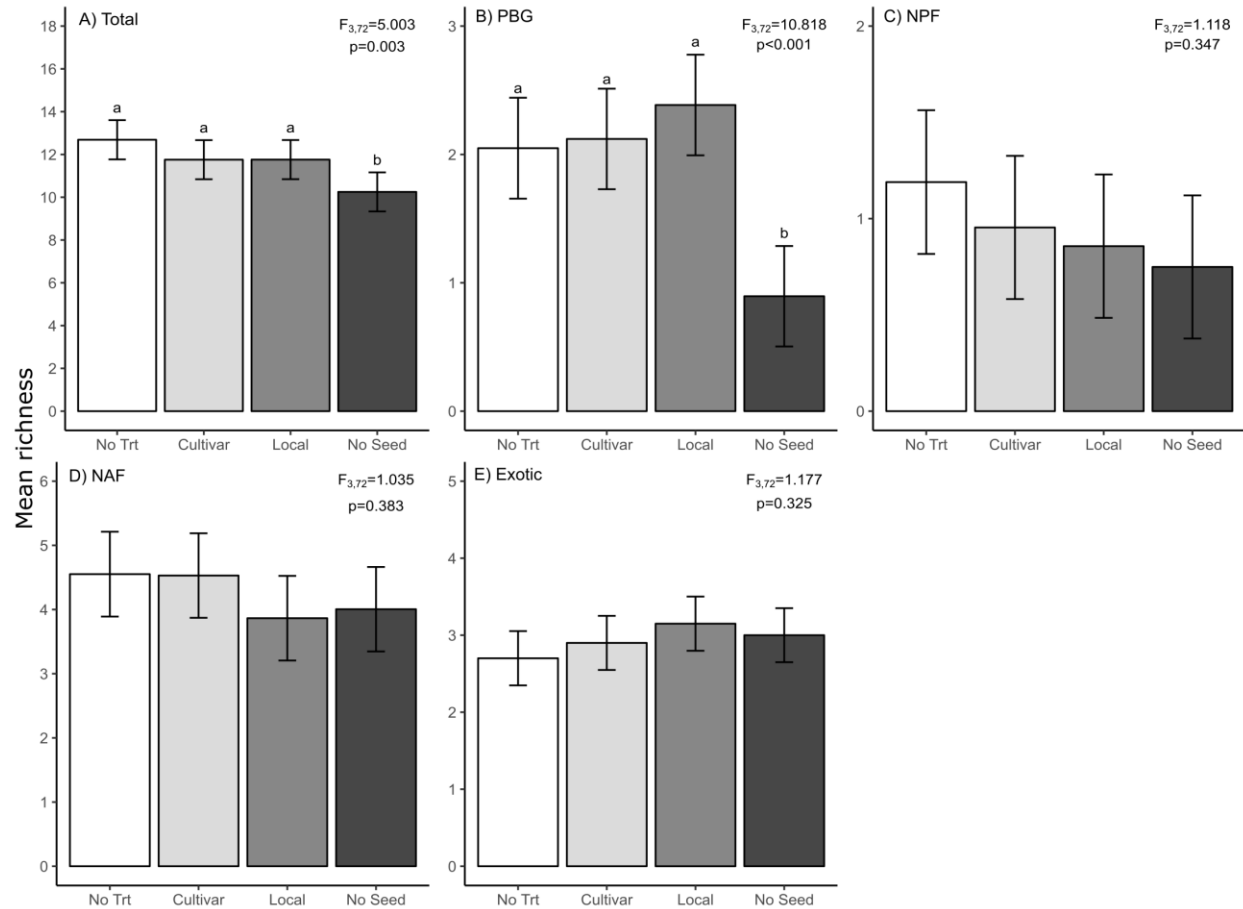
**Fig. 6.** Slash piles and functional group mean cover (%) in 2017: least square means, 95% confidence intervals and ANOVA p-values. Lowercase letters denote significance among treatments (post-hoc contrasts,  $P < 0.10$ ). Pretreatment juniper basal area was a significant covariate in the model for PSSP and EG, but did not interact with treatment.





**Fig. 7.** A slash pile plot that was not seeded and heavily invaded with cheatgrass in 2011. Results from this study suggest these areas that were not seeded have on average about 20% cheatgrass cover in 2017, which represents concerning invasion levels.





**Fig 8.** Slash piles and functional group richness in 2017: least square means, 95% confidence intervals and ANOVA p-values. Lowercase letters denote significance among treatments (post-hoc contrasts,  $P<0.10$ ). PBG = all perennial bunchgrasses, NPF = native perennial forbs, NAF = native annual forbs, Exotic = all exotics. Pretreatment juniper basal area was a significant covariate in the model for PBG ( $P = 0.07$ ).

### Skid trails

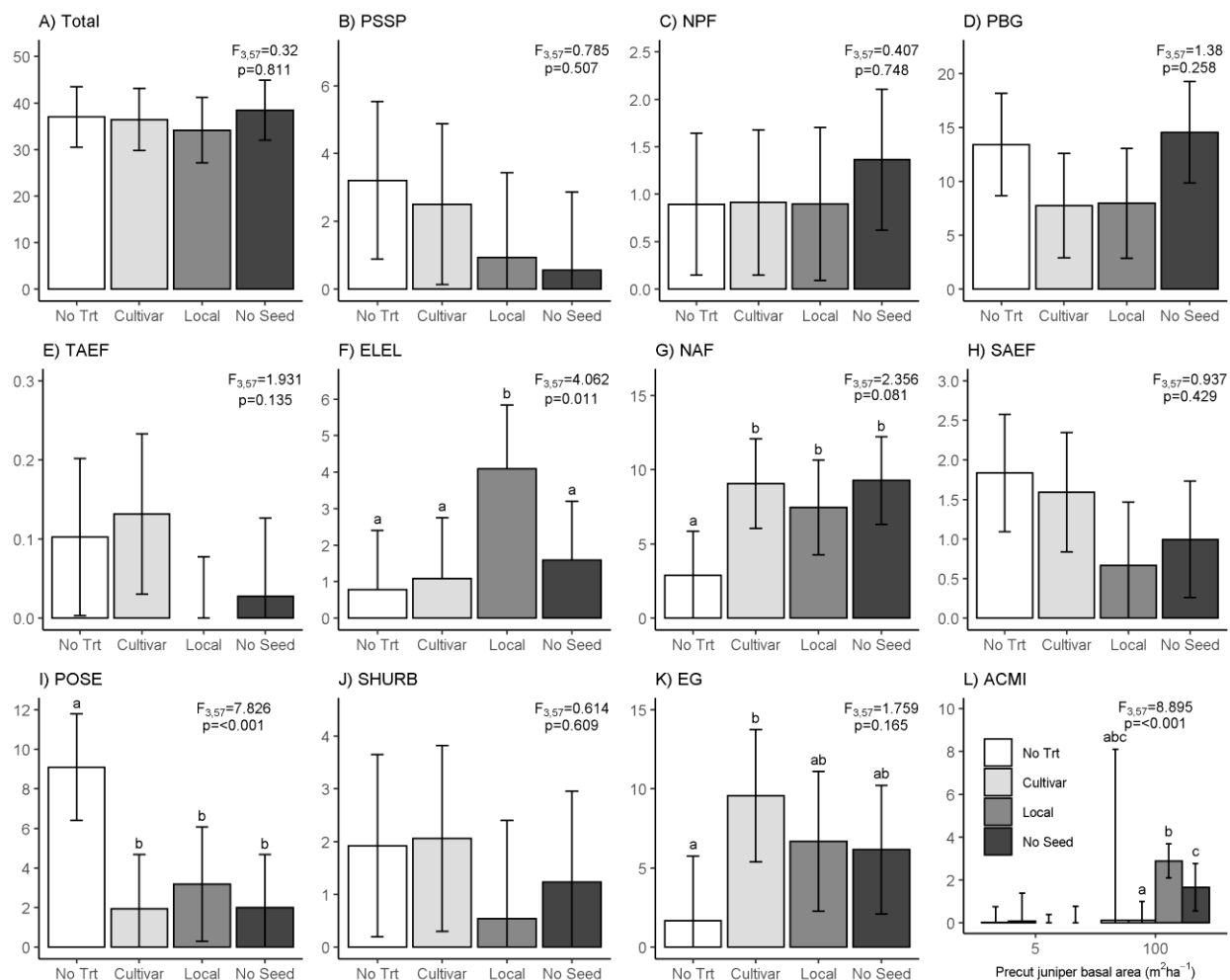
There is strong evidence total cover returned to pretreatment levels following juniper removal and skid trail formation, although there is no evidence that seeding treatments lead to higher total cover by 2017 (Fig 9A). The native perennial groups NPF and PBG showed little difference in cover compared to areas that were not treated. However, the cover for PBG was lower in plots that were seeded compared to the no seed treatment. This may suggest that seeding suppressed other native perennial bunchgrasses. There was little evidence POSE has recovered from juniper removal and skid trail disturbance although cover has increased since 2011.

There was less evidence that seeding skid trails was effective in the long-term based on cover except for squirreltail, which had 2.5 – 3.5% higher cover than no seed and no trt with the local seed mix (Fig. 9F), although we found little evidence of an effect in the short term. As mentioned, PSSP seeding on skid trails showed little effectiveness in both the short or long term.

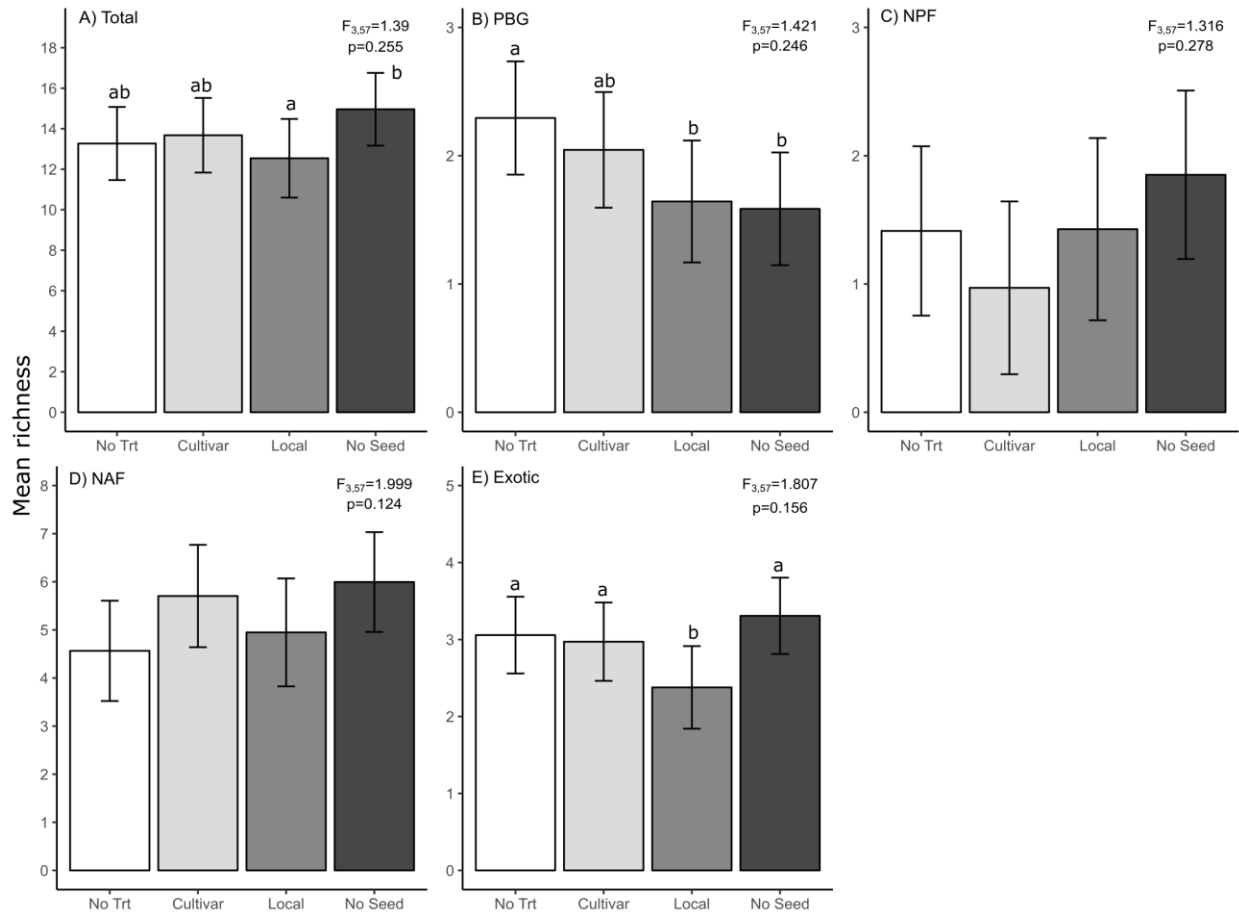
Common yarrow (ACMI) responded vigorously in the short term, but the response in 2017 depended on PJBA (Fig. 9L).

There is some evidence exotic grass cover increased in response to disturbance, and neither seed mix effectively mitigated invasion. In fact the cultivar seed mix had the highest EG cover (ca. 10%). However, it is important to note that no trt controls were nested within the larger treated landscape as shown in Fig. 3, and only the lack of seeding increased exotic grass cover above no trt but only in the short term.

Eight years after juniper removal and seven years after seeding, there is little evidence that disturbance and seeding had any effect on richness (Fig. 10), although short term differences were detected with higher richness in NPF and NAF (Kerns and Day 2014).



**Fig 9.** Functional group mean cover (%) in 2017 for skid trail experiment: least square means, 95% confidence intervals and ANOVA p-values. Lowercase letters denote significance among treatments (post-hoc contrasts,  $P < 0.10$ ). Pretreatment juniper abundance was a significant covariate for TAEF, NAF, SAEF, SHRUB and ACMI. However, the covariate only interacted with treatment for ACMI (panel L).



**Fig. 10.** Skid trail functional group mean richness in 2017: least square means, 95% confidence intervals and ANOVA p-values. Lowercase letters denote significance among treatments (post-hoc contrasts,  $P < 0.10$ ). PBG = all perennial bunchgrasses, NPF = native perennial forbs, NAF = native annual forbs, Exotic = all exotics.

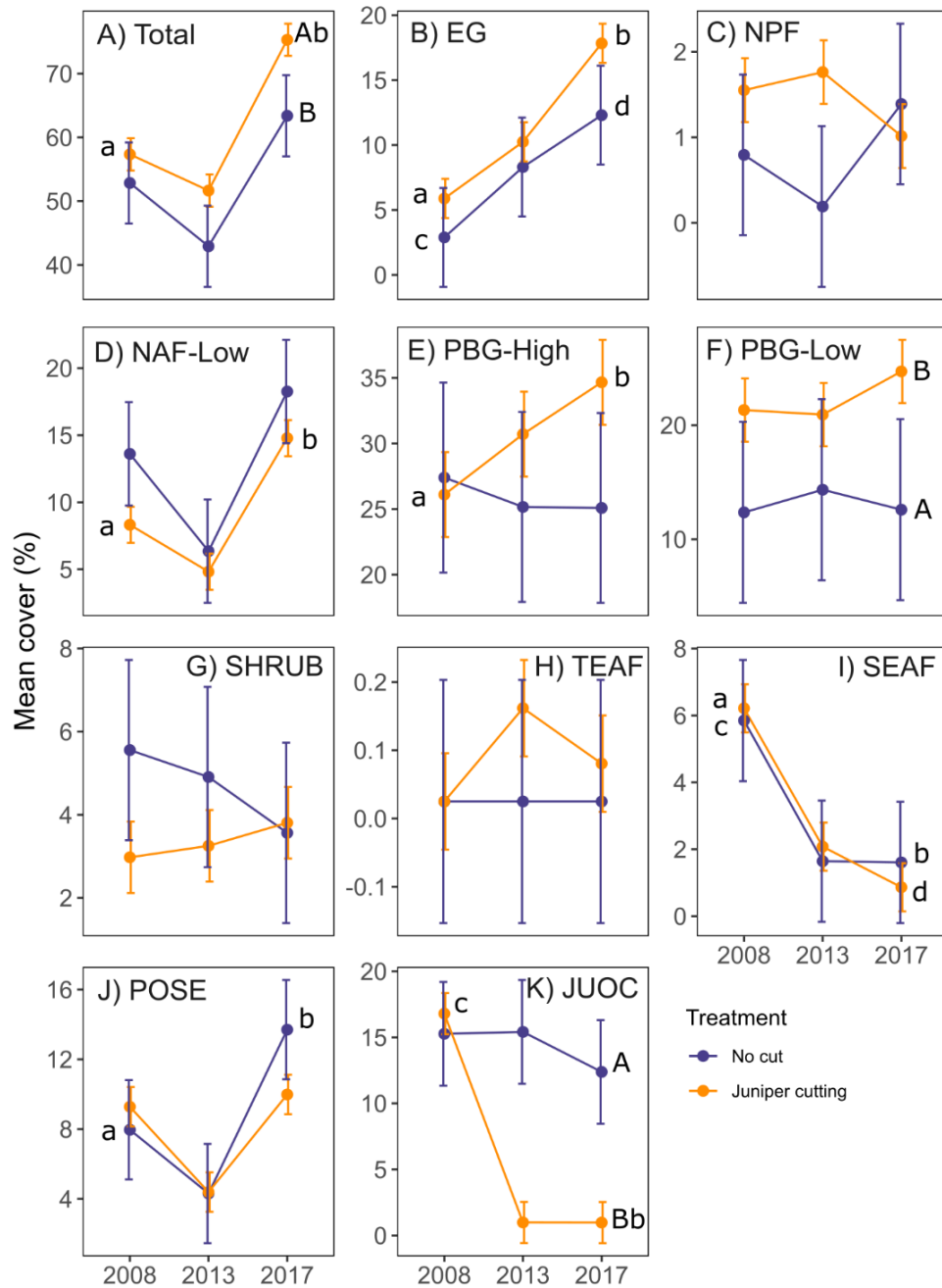
### **Long-term restoration effectiveness**

The main contrasts of interest are shown in Figure 11 from repeatedly measured transects. For all plant functional groups and total cover, there were no differences between treatment areas prior to juniper cutting. Not surprisingly, JUOC cover decreased to close to 0% from 2008 to 2017 in cut areas as the majority of trees were removed (Fig. 11). Total cover increased through time in the study area in both cut and uncut areas, but only significantly in the cut areas (18% cover increase from 2008 to 2017). Over two thirds of the increase in plant cover in the juniper cut areas was due to an increase in EG cover (12% cover increase, largely driven by cheatgrass). PBG cover also increased in cut areas, although less so than exotics (6% increase), and the response depended on pretreatment juniper cover (PJC) (Fig. 11). Where PJC was high, PBG cover increased significantly from 2008 to 2017 in cut areas, while in low PJC areas this increase was only moderate, but by 2017 PBG cover was higher in cut areas compared to uncut areas.

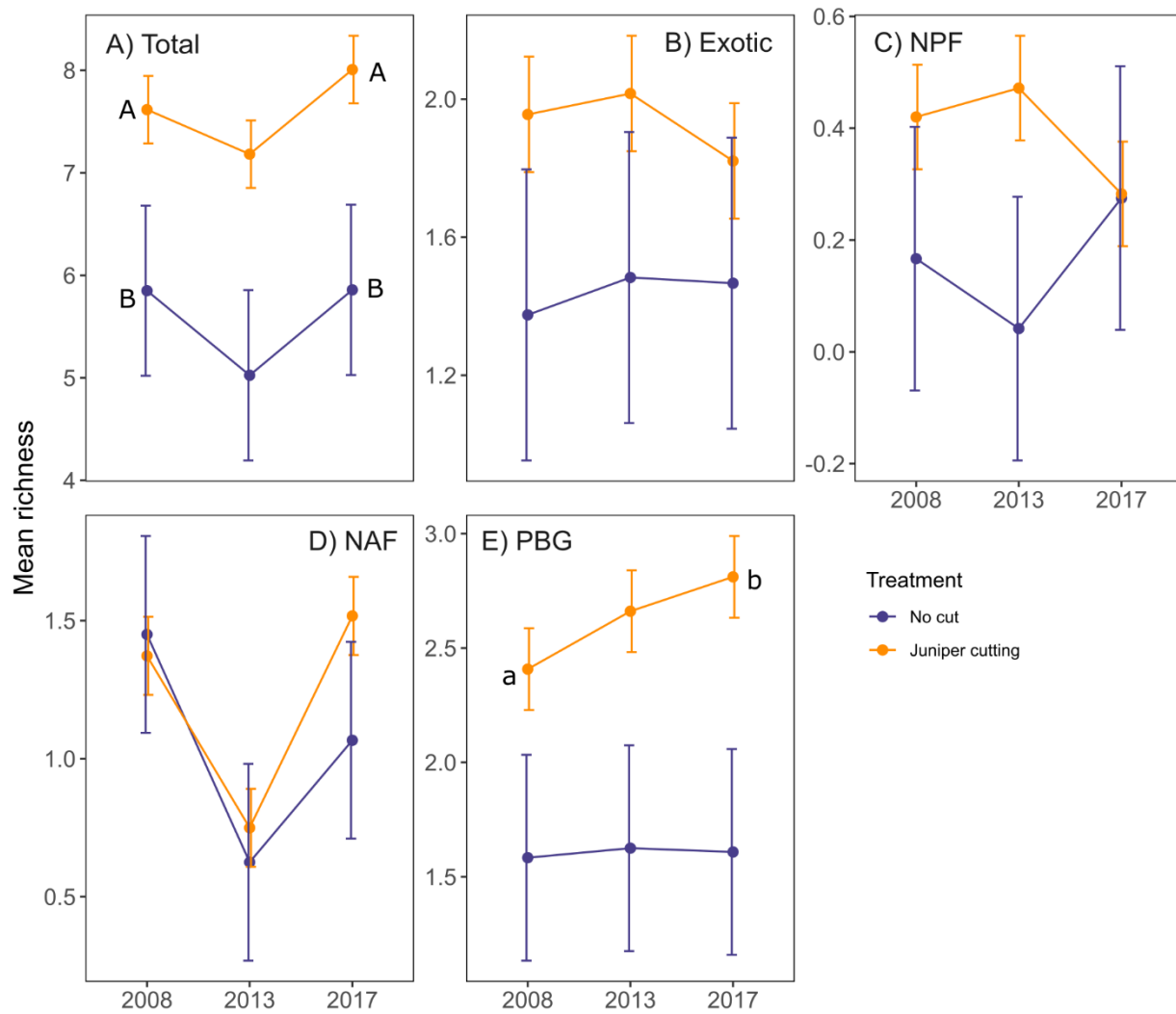
We found no evidence that juniper cutting increased shrub cover, although there was a slight increase in cover in 2017 in cut areas and a decreasing trend in uncut areas. As noted above, exotic grass (EG) cover increased in cut areas, but it also increased almost 9.5% in uncut areas. POSE cover also increased almost 6% in areas that were not cut, while SEAF cover decreased about 4-5% in both cut and uncut areas. From 2008 to 2017 NAF cover increased significantly, but only in cut areas that had low PJC (Fig. 11).

Some differences between treatments did not emerge until 2017 (total, PBG), while others were short lived and did not persist. For example, there was some evidence of slightly higher NPF cover in cut areas in 2013 ( $p = 0.030$ ; less than 2%), but the effect was gone by 2017.

For richness results, there was a significant difference in Total and PBG richness in 2008, owing to lower richness in areas selected for no cutting treatment (Fig. 12). This is not surprising as differences in plant composition in high and low density juniper areas motivated the stratification of the study site to account for potential differing responses to treatment as a result of the initial plant composition. However, given these pre-existing differences, only trends between 2008 and 2017 within each treatment for Total and PBG richness can be used for inference (Fig. 12). Total richness was not different in 2017 as compared to 2008 in either treatment, indicating juniper cutting did not change total plant richness in the long term. For PBG richness, there was a very small but significant difference in richness in 2017 (less than one species) as compared to 2008 for the areas cut. The only other difference in species group richness found was a short-lived but very small increase in NPF richness in 2013 ( $p = 0.013$ ). Taken in total, this evidence suggests that overall, juniper cutting did not increase plant species richness.



**Fig. 11.** Functional group cover through time from repeatedly measured transects in response to juniper cutting. Pretreatment data were collected in 2008. Data are least square means and 95% confidence intervals. Lowercase letters denote a significant difference between 2008 and 2017 within treatment. Uppercase letters denote significance between treatments for 2017. Other contrasts are not shown for graphical clarity, but are discussed in the text. Significant treatment interactions with pretreatment juniper cover (low or high) are shown as separate graphs.



**Fig. 12.** Functional group richness in 2017 for the long term restoration effectiveness study. Data are least square means, 95% confidence intervals and ANOVA p-values. Lowercase letters denote a significant difference between 2008 and 2017 within treatment. Uppercase letters denote significance between treatments for 2008 or 2017. PBG = all perennial bunchgrasses, NPF = native perennial forbs, NAF = native annual forbs, Exotic = all exotics.



## Key Findings and Conclusions

### Long-term Seeding Effectiveness

We remeasured and analyzed data from experimentally seeded plots in slash piles and skid trails used in our original study (Kerns and Day 2014), and assessed the long-term (~8 years) effectiveness of juniper cutting, ground disturbance, and post-treatment seeding. We report several key findings related to our research objectives.

*Seeded species persisted in the long-term, but did not facilitate broader community recovery.* Our results are similar to others showing long-term seeded species persistence (Knutson et al. 2014, Ott et al. 2019). But there was limited evidence seeding facilitated broader community recovery, and increases in species richness were very small. Increased richness due to seeding was limited to just several perennial bunchgrass species, consistent with the number of species that were seeded.

*Seeded species persistence was more evident for slash piles.* Similar to the short-term results, successfulness of seeding remained more prevalent on slash piles than skid trails. In slash piles, available nutrient levels can be elevated for several years or longer (Korb et al. 2004, Creech et al. 2012). In this study, seeded species persistence was marked for bluebunch wheatgrass, in which cover in seeded plots in 2017 was double that of either the control or no seed plots in slash piles. However, the skid trail experiment most likely has broader application to similar juniper cutting projects, as small slash pile burning may not be typically used, and small slash piles generally cover a much smaller portion of the landscape as compared to skid trails formed to remove logs. However, very large slash piles may counter this conclusion. While seeded species persistence was more evident for slash piles, exotic species cover was also higher on slash piles.

*Seeded species can help reduce exotic grass cover, even if short-term results are not promising.* There was some evidence that seeding can suppress exotic grass abundance in the long term, particularly after slash burning. Similar to other results (Rinella et al. 2012, Copeland et al. 2019, Uzra et al. 2019), this effect was not detected in the short term. Long-term results found here likely may be due to the increase in perennial bunchgrass (specifically bluebunch wheatgrass) between 2011 and 2017, as other work has demonstrated that perennial grasses can outcompete exotic annuals (Bates et al. 2000, Davies and Johnson 2017). In addition, the short-term success of seeded species common yarrow (ACMI, Kerns and Day 2014) may also have facilitated a longer-term reduction in exotic grasses.

*Cultivar and local seed sources performed differently in the long-term.* In 2017, squirreltail continued to show a strong treatment effect, with cover being twice as high in locally seeded plots relative to plots seeded with the cultivar. This pattern held true in both skid trails and slash piles. ACMI also had higher cover in slash pile areas for the local seed mix. While some managers suggest that cultivars are more aggressive and competitive compared to locally generated species (Aubrey et al. 2005), our findings do not strongly support this. Seeded species selection may be a more critical determinant of restoration success than the origin of seed mix.

*Long-term outcomes reflected successional patterns.* While short-term outcomes did not necessarily predict long-term results, temporal patterns reflected ecological succession and species life history traits. Since 2011, cover of both bluebunch wheatgrass and squirreltail doubled in seeded plots, while common yarrow decreased. This is expected given that common yarrow was selected as a “weedy” native, for its ability to spread rapidly in disturbed areas, and potential to outcompete exotic annual grasses (Alekssoff 1999), whereas squirreltail and especially bluebunch wheatgrass are both slower growing.

*Pretreatment juniper abundance had little impact on long-term plant responses.* As noted in Kerns and Day (2014), plant functional group cover and richness showed unexpected significant relationships to pre-cut juniper abundance, but pretreatment abundance did not interact with post-treatment plant response.

### **Long-term landscape scale response and restoration outcomes**

For the second study we specifically wanted to determine if key goals for sagebrush community recovery and resilience were met more generally across the landscape. Key goals that we were able to examine with this study included: increased total plant cover, and increased cover of perennial bunchgrasses, perennial forbs, and sagebrush. Another key restoration goal was to decrease the cover of shallow rooted perennials such as Sandberg bluegrass (*Poa secunda*). Similar richness metrics were also examined. Examining the vegetation response to juniper removal using our systematic transects and longitudinal data allowed us to gain a landscape perspective on the impact of these restoration treatments. However, due to the low number of transects in control areas given the variation in juniper density, our results should be interpreted with caution.

*In general, restoration treatment goals were not met.* Juniper removal increased total plant cover, but this result was largely driven by annual grass cover, resulting in concerning invasion levels that do not appear to be ephemeral. Invasion levels above 10% may be detrimental to the plant community (Ott et al. 2019), due to losses in forage species, and increase fine fuel loading. Because of the long-term increase in exotic grass cover, the restoration treatments may have actually decreased ecological resilience at this site. In addition, while removing juniper will decrease crown fire potential, exotic grasses are fine fuels that greatly increase surface fire potential and fire spread (Kerns et al. 2020), which may be problematic if fire spreads to adjacent homes, property, and heavier timber fuels.

We found no evidence that cutting alone increased the cover of shrubs. For shrubs, this was unsurprising given that sagebrush and in particular Wyoming big sagebrush can take between 5-8 years (Tausch and Tueller 1977) or longer post-disturbance to show signs of recovery (Bates et al. 2005, Bates and Davies, 2017 Schlaepfer et al. 2014). In addition, juniper removal did not decrease the cover of Sandberg bluegrass.

Cutting did increase perennial bunchgrass cover in the long term, but the response to juniper removal depended on pretreatment juniper abundance. Removing juniper in dense areas led to the largest increases in perennial bunchgrass cover. These areas already harbor perennial bunchgrasses, and have lower pretreatment cover of exotic grasses. Both factors may allow for a

vigorous understory response when cut, and challenge the notion that dense juniper areas are more prone to invasion and poor restoration outcomes as compared to less dense areas. Indeed in less dense areas with more exotic grass, cover of perennial bunchgrasses did increase after treatment, but increases were more modest as compared to more dense juniper areas. This is likely due to decreased perennial grass seed sources in more open areas and more exotic grass competition. Both factors suggest that treating exotic grasses prior to treatment in open less dense areas combined with post-treatment native seeding may improve restoration outcomes.

## **Management Implications**

*Seeded species can persist in the long-term, especially after slash pile burning, but seeding may not facilitate broader community recovery.* However, this result may be strongly controlled by initial germination success and species persistence.

*Seeded species effects on exotic grasses may not be detectable in the short term.* Although seeding may reduce exotic grass cover in the long term, invasion levels may still be problematic and have impacts to biodiversity and fire behavior. Long-term monitoring is needed to measure the impact of seeding on exotic species.

*Choosing cultivar or native seed sources might not matter.* We found limited evidence that either seed source outperformed the other in the long term, although locally sourced squirreltail was the most consistently successful species in the long term. Species selection and disturbance type were also critical determinants of success.

*Short-term results may not predict longer-term outcomes, but patterns through time were somewhat predictable based on species life history traits.* Slower growing perennials may take years to establish, and initial dominance by shorter lived species can be ephemeral. Seed mixes that include different species life history traits may provide better seeding outcomes.

*Juniper removal may not lead to restoration success or increase ecosystem resilience when exotic grasses are present.* On a landscape scale, removing juniper did increase total plant cover, and to a lesser extent perennial bunchgrass cover. Most of the increase in total cover was due to exotic annual grasses. Exotic grass cover invasion levels are concerning especially in relation to potential changes in fire behavior across this site and impacts on biodiversity and forage production.

## **Future work needed**

We see three main opportunities in which future efforts could expand upon the results reported here. First, continued monitoring would allow a great understanding of long-term restoration outcomes for juniper removal and seeding treatments. Additional monitoring would also allow us to assess the recovery of shrubs, which can take 20 years or more to fully recover. Second, an expansion of the study along environmental gradients would allow us to parse the effects of climate from successional dynamics to better inform restoration efforts. This is especially important in informing conservation and restoration efforts in the face of climate change. Lastly, plant community analyses would provide further insight into plant community responses.

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## Appendix B: Deliverables Crosswalk

List of Completed/Planned Scientific/Technical Publications/Science Delivery Products

<b>Deliverable Type</b>	<b>Description</b>
Local Presentation	Results were presented to managers in Prineville, OR, July, 2018
Refereed Publications (1)	Planned: We hope to publish results in FY20.
Conference Presentation	Results were presented at the Weed Science of Society Annual Meeting in 2020.
Workshop for Managers and Field Tour	Planned: After peer review, we will present results to forest and grassland staff and interested stakeholders with a possible field trip to treated units.
Webinar	Planned: A webinar presenting the results will be offered in coordination with the Great Basin or NW Fire Science Consortium.